

## MARSH TERRACING AS A WETLAND RESTORATION TOOL FOR CREATING FISHERY HABITAT

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**Abstract:** Terracing is a relatively new wetland-restoration technique used to convert shallow subtidal bottom to marsh. This method uses existing bottom sediments to form terraces or ridges at marsh elevation. A terrace field is constructed by arranging these ridges in some pattern that maximizes intertidal edge and minimizes fetch between ridges; the intertidal area is planted with marsh vegetation. We examined the habitat value of terracing for fishery species at Sabine National Wildlife Refuge, Louisiana (USA) in spring and fall 1999 by quantifying and comparing nekton densities in a 9-yr-old terrace field and nearby reference area using a 1-m<sup>2</sup> drop sampler. Decapod crustaceans were more abundant than fishes, composing 62% and 95% of all organisms we collected in spring and fall, respectively. White shrimp *Litopenaeus setiferus*, daggerblade grass shrimp *Palaemonetes pugio*, blue crab *Callinectes sapidus*, and brown shrimp *Farfantepenaeus aztecus* accounted for 94% of all crustaceans, whereas 60% of all fishes were gulf menhaden *Brevoortia patronus*. Mean densities of white shrimp (fall), daggerblade grass shrimp, blue crab, and brown shrimp (spring) were significantly greater in terrace marsh than on non-vegetated bottom in the reference pond. Densities of most nekton on non-vegetated bottom were similar in the terrace field and the reference pond, but gulf menhaden and white shrimp had higher densities at terrace pond sites and brown shrimp (spring) were more abundant at reference pond sites. The pattern for biomass was similar to that for density in that the mean biomass of most species was significantly greater at terrace marsh sites than reference pond sites and similar at terrace and reference pond sites. Terrace marsh, however, was not functionally equivalent to natural marsh, as mean densities of daggerblade grass shrimp (fall), brown shrimp (spring), and blue crab and mean biomass of white shrimp (fall), striped mullet *Mugil cephalus* (spring), and spotted seatrout *Cynoscion nebulosus* (fall) were greater at reference marsh sites than terrace marsh sites. Using these density and biomass patterns and the percentage of marsh and pond area in the terrace field, we concluded that terrace fields support higher standing crops of most fishery species compared with shallow marsh ponds of similar size. Future restoration projects could include design changes to increase the proportion of marsh in a terrace field and enhance the habitat value of marsh terraces for fishery species.

**Key Words:** fishery species, Gulf of Mexico, habitat comparisons, habitat selection, nursery areas, penaeid shrimps, tidal marsh, restoration

### INTRODUCTION

Estuarine marshes, and in particular the vegetation at the marsh-water interface (marsh edge), harbor dense concentrations of young fishery species (Zimmerman and Minello 1984, Baltz et al. 1993, Cicchetti 1998, Howe et al. 1999, Rozas and Zimmerman 2000). As nursery areas, these wetlands support valuable coastal fisheries along the Gulf of Mexico and Southeast Atlantic coasts of the U.S. (Kneib 1997, Zimmerman et al. 2000). Over the last 50 years, the total area of coastal marsh habitat has decreased substantially as wetlands were filled for development or converted to open water through submergence (Mitsch and Gosselink 1986).

In response to this large-scale loss of coastal wetlands, a variety of techniques have been used to restore these estuarine habitats. Structural marsh management, or the use of impoundments with water-control structures, has been used as a wetland restoration technique that may be effective in restoring wetland vegetation in certain oligohaline areas (see Boyer 1997 for an evaluation of effectiveness), but evidence is strong that this approach has negative effects on habitat use by transient fishery species (McGovern and Wenner 1990, Rogers et al. 1994, Rozas and Minello 1999). In direct contrast, wetland restoration through re-establishment of tidal flow to diked marsh appears to positively affect habitat use by fishery species (Gilmore et al. 1982, Rey et al. 1990, Chamberlain and Barnhart 1993, Peck et

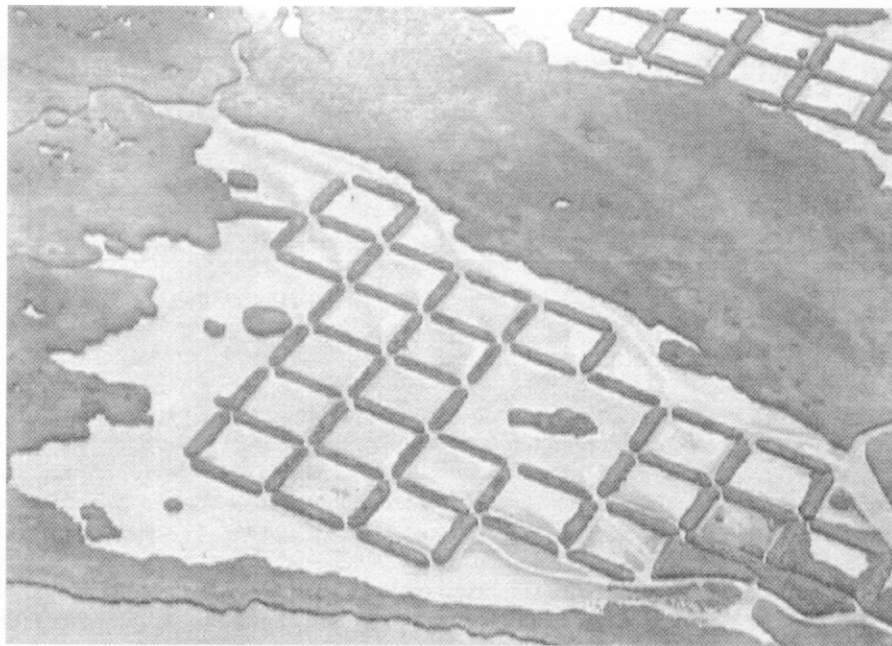


Figure 1. Aerial photograph of a terrace field at Sabine National Wildlife Refuge showing terraces laid out in a checkerboard pattern. Corners are open to allow movement of water and organisms through the terrace field. (Photograph taken January 16, 2001 by Ron Boustany and Tommy Michot.)

al. 1994). Perhaps the most common restoration technique, however, simply involves planting marsh vegetation on dredged material or on scraped-down uplands. In the southeastern United States, this approach appears to restore some habitat functions for estuarine nekton, but there is also evidence that these created marshes are not functionally equivalent to natural marshes (Moy and Levin 1991, Minello and Zimmerman 1992, Meyer et al. 1993, Minello and Webb 1997, Minello 2000, Streever 2000). One important reason for this deficiency is that these created marshes (especially those constructed on fine-grained dredged material) have relatively few creeks and ponds, and consequently little edge habitat (Minello et al. 1994, Delaney et al. 2000, Streever 2000).

Marsh terracing is a relatively new wetland-restoration technique that creates a substantial amount of marsh edge (Underwood et al. 1991). Terraces are ridges or levees of sediment at marsh elevation that are constructed by excavating subtidal bottom sediments on site. The borrow areas created by terrace construction trap and eventually become filled with sediment. Terrace fields are developed by arranging a series of these ridges in some pattern that maximizes intertidal edge and minimizes fetch between ridges. One arrangement is a checkerboard pattern with open corners (Figure 1). Following construction, the intertidal levees are planted with marsh vegetation; and the subtidal areas between terrace levees, may be planted with seagrasses or other species of submerged aquatic

vegetation (SAV). Marsh terracing has been promoted as a means of enhancing deposition and retention of suspended sediments, reducing turbidity, increasing marsh-edge habitat, increasing overall primary and secondary productivity, and maximizing access for marine organisms (Underwood et al. 1991, LDNR 1993). Currently, marsh terracing appears to be restricted to restoration efforts along the northern Gulf of Mexico coast. However, the technique could be applied in other areas with shallow water and suitable substrates. For example, land-reclamation methods used for decades in Europe are quite similar to the techniques used in marsh terracing (Kamps 1962).

The first terracing project in the U.S. was constructed at the Sabine National Wildlife Refuge (Sabine NWR), Cameron Parish (county), Louisiana in 1990 (Underwood et al. 1991, LDNR 1993). *Spartina alterniflora* Loisel. was successfully established along the terrace levees in October 1990, and this species dominated well-established stands of marsh vegetation on the terraces when we initiated our study in 1999. An initial attempt to establish SAV was apparently unsuccessful; three species of SAV (*Ruppia maritima* Linnaeus, *Halodule wrightii* Ascherson, and *Thalassia testudinum* König) were planted in experimental plots within the newly-constructed terrace ponds in June 1991, but few plants had survived by August 1991 (LDNR 1993).

A major goal of marsh terracing is to provide habitat for fishery species, but the habitat value of terrace

fields has not been assessed. Specific objectives of our study were to document the use of marsh terraces by nekton (dominant species of fishes and decapod crustaceans) and to compare densities and biomasses of fishery species (as a measure of habitat quality) in a terraced area and a nearby reference area at Sabine NWR. Our use of animal density and biomass to measure habitat quality is predicated on the assumption that animal distributions reflect the capacity of an area to provide essential habitat functions for species. The results from our research can be used to predict changes in habitat value that may be anticipated in areas where new terracing projects are now being planned.

## MATERIALS AND METHODS

### Study Area

The study area is in Southwest Louisiana within the Sabine NWR and located on Calcasieu Lake (Figure 2). Salinities at the study area are 10–15‰ in spring and 15–20‰ in fall during years of average rainfall (Orlando et al. 1993). Tides in the estuary are predominantly diurnal and have a mean daily range of <0.3 m (U.S. Department of Commerce 1998). Salt marsh dominated by *Spartina alterniflora* occupies much of the low intertidal zone within the study area.

Terrace fields were built in 1990 within two of three shallow marsh ponds located on the north shore of West Cove, an arm of Calcasieu Lake (Figure 2). These terrace fields were constructed in a checkerboard pattern with cells having 61-m-long sides (0.37-ha pond area). Intertidal terrace ridges within cells were approximately 55 m long. We selected the pond without terraces as a reference area and the adjacent terraced pond as the two areas for comparison in our study design (Figure 2). We identified four treatments for sampling: two treatments were intertidal and vegetated with *Spartina alterniflora* (terrace marsh and reference marsh) and two were subtidal open-water, mud bottom (terrace pond and reference pond). Most of the shallow, open-water areas did not support SAV, but a few areas contained *Ruppia maritima*.

In each of two seasons (spring and fall), we collected a total of 60 samples from randomly selected sites at the terrace and reference areas; the sites were selected using random numbers and a grid placed over an aerial photograph. We took ten *Spartina* marsh edge samples each on terrace levees (terrace marsh) and along the shoreline of the reference pond (reference marsh). In addition, 20 samples each were taken in terrace ponds and the reference pond. We collected twice as many open-water samples as marsh samples to ensure that at least some sites would contain SAV. Further, more samples are required to accurately rep-

resent non-vegetated areas because these sites contain relatively few organisms and variability among sites is high. We collected all samples in the day at high tide during periods of tropical tides May 4–5 and September 7–8 and 20, 1999.

### Nekton Sampling

Fishes and decapod crustaceans (nekton) were quantitatively sampled with a drop sampler using the procedure described by Zimmerman et al. (1984). We chose a drop sampler for this study because it is effective in dense emergent vegetation, and the catch efficiency of this enclosure device does not appear to vary substantially with habitat characteristics typical of shallow estuarine areas (Rozas and Minello 1997). The sampler was a 1.14-m-diameter cylinder that we dropped from a boom attached to a shallow-draft boat. Two persons positioned the cylinder over a sample site by slowly pushing from the boat's stern. When released from the boom, the cylinder rapidly entrapped organisms within a 1.0-m<sup>2</sup> sample area.

After the cylinder was dropped, we measured water temperature, dissolved oxygen, salinity, and turbidity using the methods described by Minello and Zimmerman (1992). We determined water depth at each sample site by averaging five depth measurements taken within the sampler. We also measured the distance from the sample area to the nearest marsh-water interface. At marsh sites, plant stems were clipped at ground level, counted (dead and alive combined), and removed from the cylinder. If SAV was present at terrace or reference pond sites, we estimated coverage within the sampler (0–100%) and identified the species of plants present.

After measuring the environmental variables, we captured nekton trapped in the drop sampler by using dip nets and filtering the water pumped out of the enclosure through a 1-mm-mesh net. When the sampler was completely drained, any animals remaining on the bottom were removed by hand. Samples were preserved in formalin and returned to the laboratory for processing.

In the laboratory, the samples were sorted, and animals were identified to lowest feasible taxon. We used the nomenclature of Perez-Farfante and Kensley (1997) for penaeid shrimps and identified species using the protocol described in Rozas and Minello (1998). Thirty-one specimens of *Farfantepenaeus* could not be reliably identified either because of their size (total length 13–18 mm) or because they were damaged; these shrimps were assigned as brown shrimp *F. aztecus* (Ives) or pink shrimp *F. duorarum* (Burkenroad) based on the proportion of identified species in each sample. An additional 79 damaged

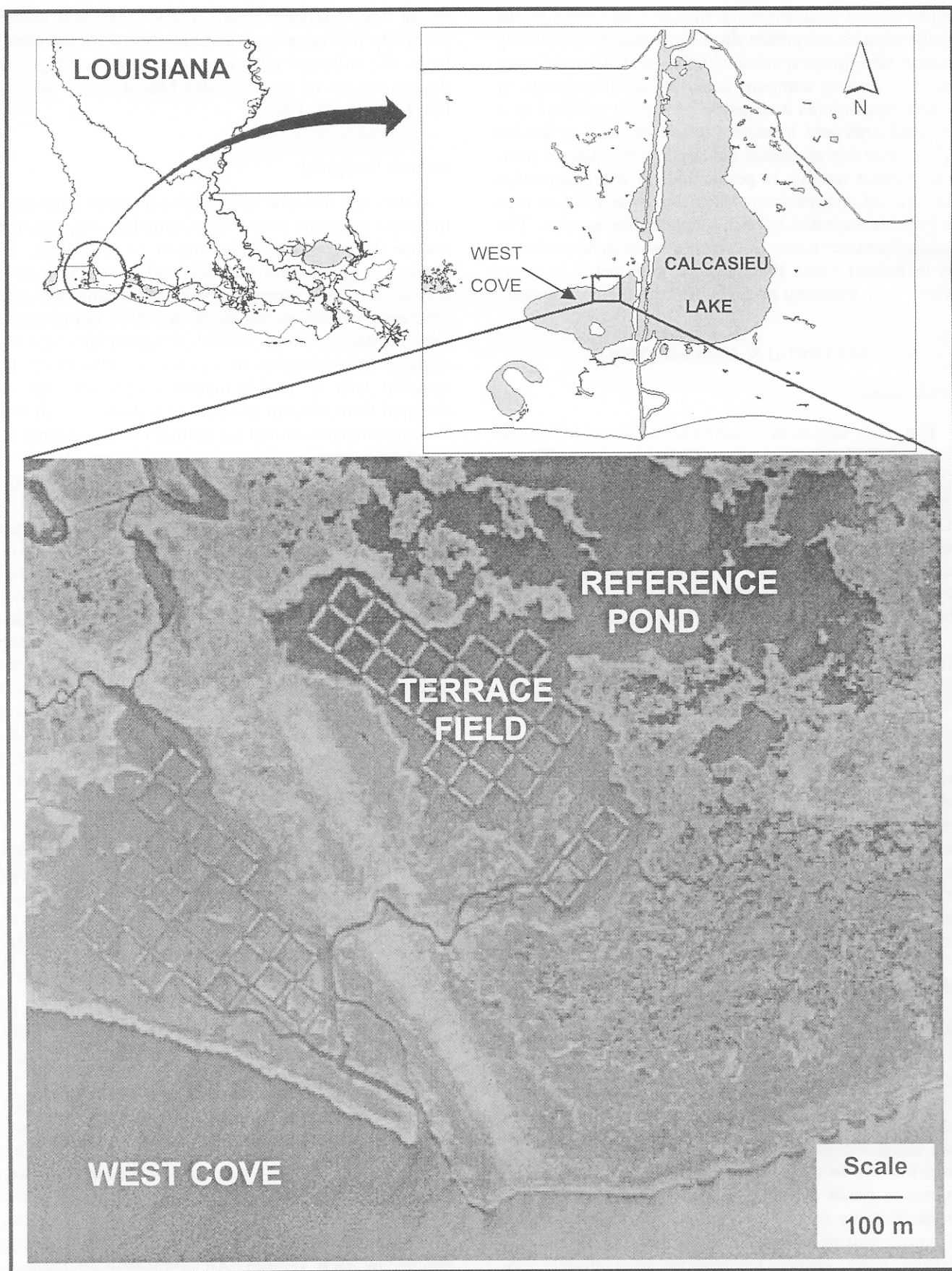


Table 1. Analysis of Variance (ANOVA) table for comparing the four treatment areas. Model includes the test for the main effect of Treatment and the *a priori* contrasts that compare specific treatment effects. The dependent variable used in the example presented here is total macrofauna (sum of total fishes and crustaceans).

Source	df	Sum of Squares	Mean Square	F Value	P Value
<i>May 1999</i>					
TREATMENT	3	58.451	19.484	17.011	0.0001
CONTRASTS					
Terrace Marsh vs Reference Pond	1	29.868	29.868	26.076	0.0001
Terrace Pond vs Reference Pond	1	0.052	0.052	0.046	0.8316
Terrace Marsh vs Reference Marsh	1	0.003	0.003	0.003	0.9592
RESIDUAL ERROR	56	64.142	1.145		
<i>September 1999</i>					
TREATMENT	3	55.395	18.465	57.136	0.0001
CONTRASTS					
Terrace Marsh vs Reference Pond	1	25.844	25.844	79.969	0.0001
Terrace Pond vs Reference Pond	1	2.352	2.352	7.277	0.0092
Terrace Marsh vs Reference Marsh	1	1.204	1.204	3.725	0.0587
RESIDUAL ERROR	56	18.098	0.323		

shrimp could only be identified as penaeids and were assigned to species (white shrimp *Litopenaeus setiferus* (Linnaeus), brown shrimp, or pink shrimp) using this same procedure. Grass shrimp (146) that could not be identified to species were similarly assigned to one of two species (daggerblade grass shrimp *Palaemonetes pugio* Holthuis, or brackish grass shrimp *P. intermedius* Holthuis) identified from our samples. Animals that could not be reliably identified, either due to their size or because they were damaged, were not used in size analyses. Total length of fishes and shrimps and carapace width of crabs were measured to the nearest mm. The biomass for each species was determined by pooling individuals in a sample and measuring wet weight to the nearest 0.1g.

As a wetland restoration technique, terrace fields are built to replace shallow open water in marsh ponds. Therefore, we also used our density data to compare standing crops (in numbers of organisms and biomass for selected taxa) between a 1-ha terrace field and a reference pond of the same size. We used recent aerial photography to calculate that 21% of the terraced area was occupied by marsh and 79% by pond. We recognize that a portion of each terrace ridge may have been exploited very little by aquatic organisms because the middle of the ridge was relatively high in elevation. For this exercise, however, we assumed that the entire terrace ridge was equally used by fishery species. We estimated standing crops of animals within

the two 1-ha areas for May and September using average animal densities and biomass determined from our nekton samples and the total area of each habitat type within the terrace field and reference pond. We also used these data to simulate the effect of increasing the proportion of marsh in a terrace field on brown shrimp standing crops (total numbers) in May.

#### Data Analyses

We used 1-way Analysis of Variance (ANOVA) followed by *a priori* contrasts to examine differences in densities and biomass of selected organisms and environmental characteristics (mean dissolved oxygen, salinity, water temperature, turbidity, water depth, and distance to edge) among treatments (terrace marsh, reference marsh, terrace pond, and reference pond) (Table 1). We compared the following treatments with *a priori* contrasts (Table 1): Terrace Marsh vs Reference Pond, Terrace Pond vs Reference Pond, and Terrace Marsh vs Reference Marsh. These first two contrasts were selected because terrace marsh and terrace pond are habitat types created from the reference pond habitat type when terraces are constructed. The terrace marsh-reference marsh contrast was used to determine whether created marsh in the terrace field was comparable to natural marsh in supporting nekton.

In the ANOVA procedure, we analyzed the data collected during each season separately, because several

Figure 2. Map of the study area and its location on the north shore of West Cove in Southwest Louisiana. Study area is located within the Calcasieu Lake estuary at approximately 29° 53' N latitude and 93° 23' W longitude.



species were only abundant enough to include in the statistical analysis in one season. We considered alpha levels of 0.05 to be significant in all results, but we also assessed significance after adjusting alpha levels for the Treatment effect using the sequential Bonferroni method described by Rice (1989), which buffers against error introduced by making multiple comparisons with the same sample set (i.e., testing a hypothesis for several species or variables). Densities and biomass of animals were positively related to the standard deviation; therefore, we performed a  $\ln(x + 1)$  transformation of the original values prior to analyses. Other variables were not transformed. All tabular and graphical data presented in this paper are untransformed means. We conducted statistical analyses using SuperANOVA (Version 5 Ed., Abacus Concepts, Inc., Berkeley, California, 1989).

## RESULTS

We collected a total of 1,344 animals (21 fish and 9 decapod crustacean species) with a biomass of 2.21 kg wet weight in May and 3,575 animals (18 fish and 12 decapod crustacean species) and 0.8 kg wet weight in September (Tables 2 and 3). Fishes accounted for 58% of the total biomass in May, whereas decapod crustaceans composed most (83%) of the biomass in September. Crustaceans outnumbered fishes in both sampling periods, accounting for 62% and 95% of all organisms taken in May and September, respectively. Species of decapod crustaceans in our samples were relatively consistent between sampling periods (Table 2). Daggerblade grass shrimp, brown shrimp, blue crab *Callinectes sapidus* Rathbun, brackish grass shrimp, and pink shrimp accounted for 98% of the crustaceans taken in May. These same species, along with white shrimp, composed 99% of the crustaceans collected in September. Other taxa infrequently taken in our samples included the bigclaw snapping shrimp *Alpheus heterochaelis* Say, the lesser blue crab *Callinectes similis* Williams, the glabrous broadface crab *Euryplax nitida* Stimpson, the xanthid crabs *Rhithropanopeus harrisi* (Gould) and *Dyspanopeus texana* (Stimpson), and the fiddler crabs *Uca longisignalis* Salmon and *Uca rapax* (Smith).

Abundant fishes differed between May and September (Table 2). Most (86% of total) fishes taken in May were gulf menhaden *Brevoortia patronus* Goode and spot *Leiostomus xanthurus* Lacepede, whereas naked goby *Gobiosoma bosc* (Lacepede), code goby *Gobiosoma robustum* Ginsburg, and unidentified gobies (Family Gobiidae) were most abundant in September. Other less abundant but commonly collected species included *Anchoa mitchilli* (Valenciennes), *Cynoscion nebulosus* (Cuvier), *Cyprinodon variegatus* Lacepede,

*Fundulus grandis* Baird and Girard, *Gobiesox strumosus* Cope, *Gobionellus boleosoma* (Jordan and Gilbert), *Menidia beryllina* (Cope), *Micropogonias undulatus* (Linnaeus), and *Mugil cephalus* Linnaeus. In addition, we infrequently collected (<5 individuals) *Adinia xenica* (Jordan and Gilbert), *Citharichthys spilopterus* Günther, *Cynoscion arenarius* Ginsburg, *Etropus crossotus* Jordan and Gilbert, *Evorthodus lyricus* (Girard), *Fundulus pulvereus* (Evermann), *Fundulus similis* (Baird and Girard), *Gobionellus oceanicus* (Pallas), *Harengula jaguana* Poey, *Lagodon rhomboides* (Linnaeus), *Lucania parva* (Baird and Girard), *Mugil curema* Valenciennes, *Myrophis punctatus* Lütken, *Paralichthys lethostigma* Jordan and Gilbert, *Poecilia latipinna* (Lesueur), *Sciaenops ocellatus* (Linnaeus), *Stellifer lanceolatus* (Holbrook), and *Symphurus plagiatus* (Linnaeus) in the study area.

Mean densities of decapod crustaceans were greatest in intertidal marsh, and most species were significantly more abundant at terrace marsh sites than reference pond sites (Table 2). Exceptions were brown shrimp (in September only) and brackish grass shrimp; densities at terrace marsh sites and reference pond sites for these species were not significantly different. Crustacean densities were generally low over all non-vegetated bottom, and mean densities of most species were not significantly different between terrace pond and reference pond sites. Exceptions to this generality were brown shrimp (in May), which were more abundant at reference pond sites, and white shrimp (in September), which were more abundant in terrace ponds. Four crustacean species had mean densities in the terrace marsh that were only 11–54% of their densities in the reference marsh (brown shrimp=19% and 54%, blue crab=32% and 44%, brackish grass shrimp=11% and 18%, and daggerblade grass shrimp in September=54%). Densities of daggerblade grass shrimp in May and white shrimp in September, however, were not significantly different between marsh treatments. Fishes did not follow such a consistent spatial pattern. Gulf menhaden was most abundant over non-vegetated bottom, and mean densities of this species were greater at terrace pond than reference pond sites (Table 2). Mean densities of naked goby and code goby were greatest in the reference marsh; however, most unidentified gobies were taken in the reference pond (Table 2).

The pattern for biomass was similar to the density pattern for most numerically-dominant species (Table 3). Biomass and density patterns differed, however, for brown shrimp, blue crab, and white shrimp. Mean biomass of brown shrimp (May) was greater at terrace marsh sites than reference pond sites and did not differ significantly between terrace pond sites and reference pond sites (Table 3). Mean blue crab biomass at ref-

Table 2. Comparison of densities (mean  $m^{-2} \pm 1$  SE) of the most abundant decapod crustaceans and fishes collected among treatments (terrace marsh, reference marsh, terrace pond, and reference pond) in May and September 1999. Each mean is estimated from 10 (marsh) or 20 (pond) samples. The total number of species collected in each taxonomic category also is given. Results (p values) are given for ANOVA analyses we used to compare mean densities among the four treatment areas and *a priori* contrasts testing for significant differences between: 1 = terrace marsh and reference pond; 2 = terrace pond and reference pond; and 3 = terrace marsh and reference marsh. The ANOVA model used to do these analyses is shown in Table 1. An \* indicates that the probability value was significant for the overall ANOVA after alpha was adjusted as described by Rice (1989).

Species	Reference								Contrast p Values					
	Terrace Marsh		Reference Marsh		Terrace Pond		Reference Pond		(1) T Marsh vs R Pond	(2) T Pond vs R Pond	(3) T Marsh vs R Marsh			
	Mean	SE	Mean	SE	Mean	SE	Mean	SE						
<i>May 1999</i>														
Crustaceans (Total = 9 species)														
Daggerblade grass shrimp	26.6	(6.52)	16.0	(2.47)	0.0	(0.00)	0.2	(0.11)	429	51.3	0.0001*	0.0001	0.5735	0.2947
Brown shrimp	4.6	(0.40)	8.5	(1.15)	0.6	(0.17)	1.6	(0.50)	174	20.8	0.0001*	0.0001	0.0096	0.0321
Blue crab	3.1	(0.53)	7.0	(1.59)	0.2	(0.08)	0.5	(0.20)	114	13.6	0.0001*	0.0001	0.2790	0.0324
Brackish grass shrimp	1.3	(0.91)	7.4	(3.54)	0.0	(0.00)	0.3	(0.30)	93	11.1	0.0006*	0.2241	0.6691	0.0275
Pink shrimp	0.8	(0.59)	0.3	(0.30)	0.0	(0.00)	0.1	(0.05)	13	1.6				
Total Crustaceans	37.1	(6.81)	39.7	(4.94)	0.8	(0.19)	2.7	(1.01)	837		0.0001*	0.0001	0.0066	0.5069
Fishes (Total = 21 species)														
Gulf menhaden	2.6	(1.87)	0.0	(0.00)	15.3	(8.00)	4.4	(3.99)	418	82.4	0.0368	0.5378	0.0221	0.2381
Spot	0.8	(0.49)	0.2	(0.13)	0.2	(0.08)	0.3	(0.16)	19	3.7				
Total Fishes	5.5	(2.32)	2.1	(0.43)	15.9	(8.05)	5.7	(3.98)	507		0.5780			
<i>September 1999</i>														
Crustaceans (Total = 12 species)														
White shrimp	62.1	(11.92)	64.2	(8.84)	19.6	(2.67)	7.8	(1.11)	1810	53.5	0.0001*	0.0001	0.0006	0.6441
Daggerblade grass shrimp	38.0	(7.84)	70.8	(13.02)	0.4	(0.22)	0.2	(0.08)	1098	32.4	0.0001*	0.0001	0.6551	0.0023
Blue crab	4.7	(0.65)	14.6	(1.56)	2.6	(0.44)	2.0	(0.35)	285	8.4	0.0001*	0.0017	0.3308	0.0001
Brackish grass shrimp	0.8	(0.70)	7.0	(3.77)	0.6	(0.60)	0.0	(0.00)	90	2.7	0.0024*	0.3295	0.5795	0.0151
Brown shrimp	0.6	(0.40)	3.1	(0.80)	0.2	(0.16)	0.6	(0.20)	52	1.5	0.0001*	0.9236	0.1721	0.0001
Pink shrimp	0.0	(0.00)	1.4	(1.40)	0.2	(0.08)	0.2	(0.20)	22					
Total Crustaceans	107.2	(18.08)	162.4	(19.22)	23.5	(3.15)	10.9	(1.45)	3384		0.0001*	0.0001	0.0013	0.0699
Fishes (Total 18 species)														
Unidentified goby	0.1	(0.10)	0.5	(0.27)	0.2	(0.12)	1.5	(0.56)	39	20.4	0.0082*	0.0058	0.0026	0.3219
Naked goby	0.0	(0.00)	0.9	(0.46)	0.3	(0.30)	0.6	(0.36)	26	13.6	0.1695			
Code goby	0.0	(0.00)	1.0	(0.67)	0.0	(0.00)	0.7	(0.30)	23	12.0	0.0657			
Gulf killifish	1.1	(0.50)	0.2	(0.20)	0.0	(0.00)	0.0	(0.00)	13	6.8				
Total Fishes	3.1	(0.97)	4.9	(0.94)	1.8	(0.81)	3.8	(0.68)	191		0.0007*	0.3853	0.0009	0.1051





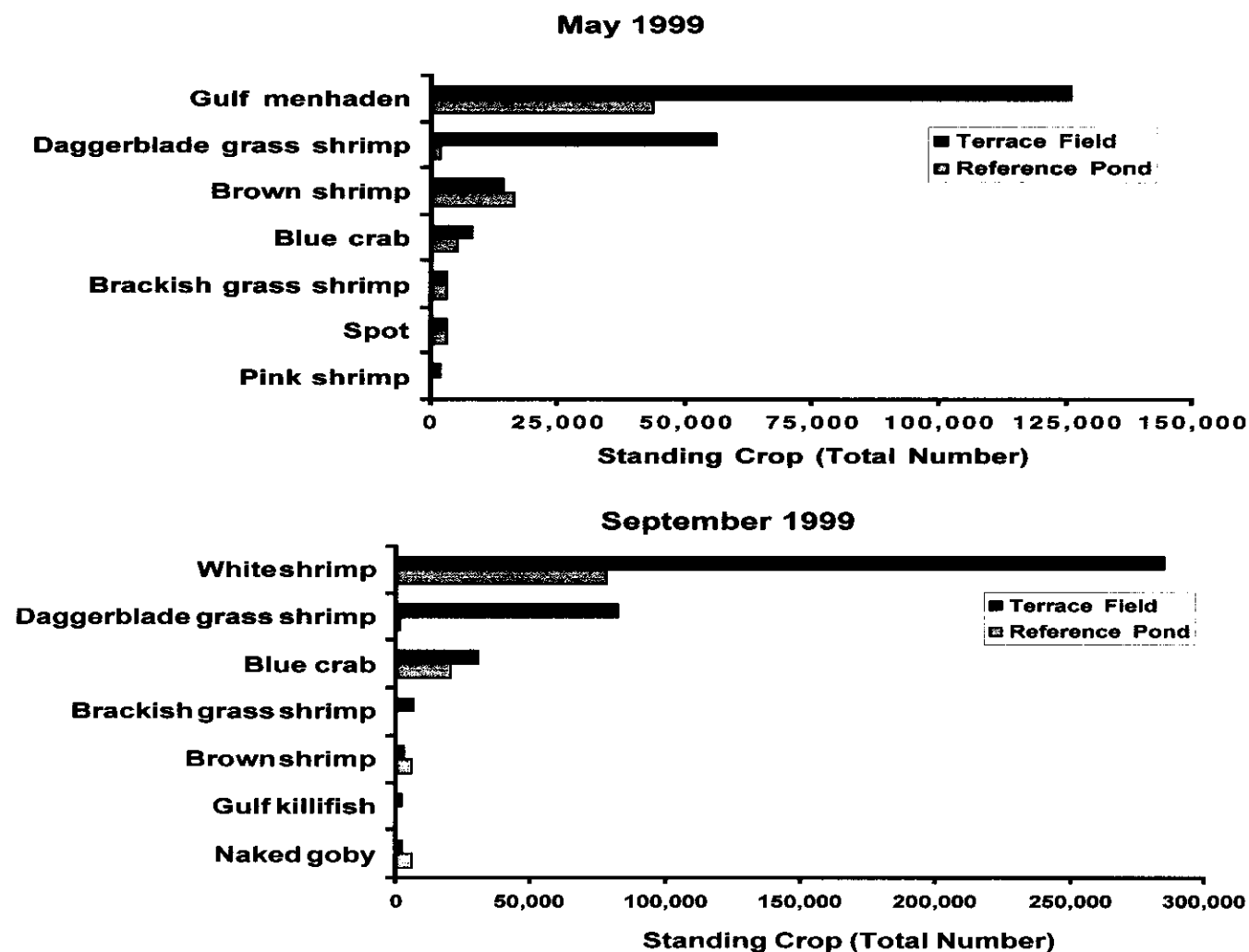


Figure 3. Comparison of hypothetical 1-ha terrace field and reference pond in May and September 1999 using estimated standing crop (total number of organisms) of abundant taxa. Animal densities used in the calculations are mean densities determined from sample collections in each habitat type (terrace marsh and terrace pond, reference pond). Design of the terrace field in this comparison is based on the marsh terraces in our study area where marsh and ponds occupy 21% and 79% of the total area, respectively.

erence marsh sites and terrace marsh sites was not significantly different, and mean biomass of white shrimp (September) was significantly greater at reference marsh sites than terrace marsh sites (Table 3). The reference marsh contributed most of the biomass of three fish species (striped mullet, white mullet, and spotted seatrout) that accounted for a substantial percentage of the total biomass, even though they were not abundant in our samples (Table 3). The mean biomass of striped mullet was significantly greater at reference marsh sites than terrace marsh sites in May, although this species was collected exclusively at terrace marsh sites in September (Table 3). Spotted seatrout was taken almost exclusively in the reference marsh, and white mullet was collected only in the reference marsh (Table 3).

In comparisons between 1-ha areas of marsh terrace and reference pond, standing crops of most animals were greater within the terrace field (Figures 3 and 4). Based on our density data for May, gulf menhaden, daggerblade grass shrimp, blue crab, and pink shrimp were 2.9, 37.2, 1.5, and 3.4 times more abundant, respectively, in the terrace field than the reference pond. In September, white shrimp, daggerblade grass shrimp, and blue crab were 3.7, 55.0, and 1.5 times more abundant, respectively, in the terrace field. In contrast, standing crops (total numbers) of brown shrimp in the terrace field in May were 88% of those in the reference pond (Figure 3). Naked goby also was less abundant in the terrace field than in the reference pond. Comparisons of standing crops using biomass data followed a similar pattern (Figure 4). The terrace field

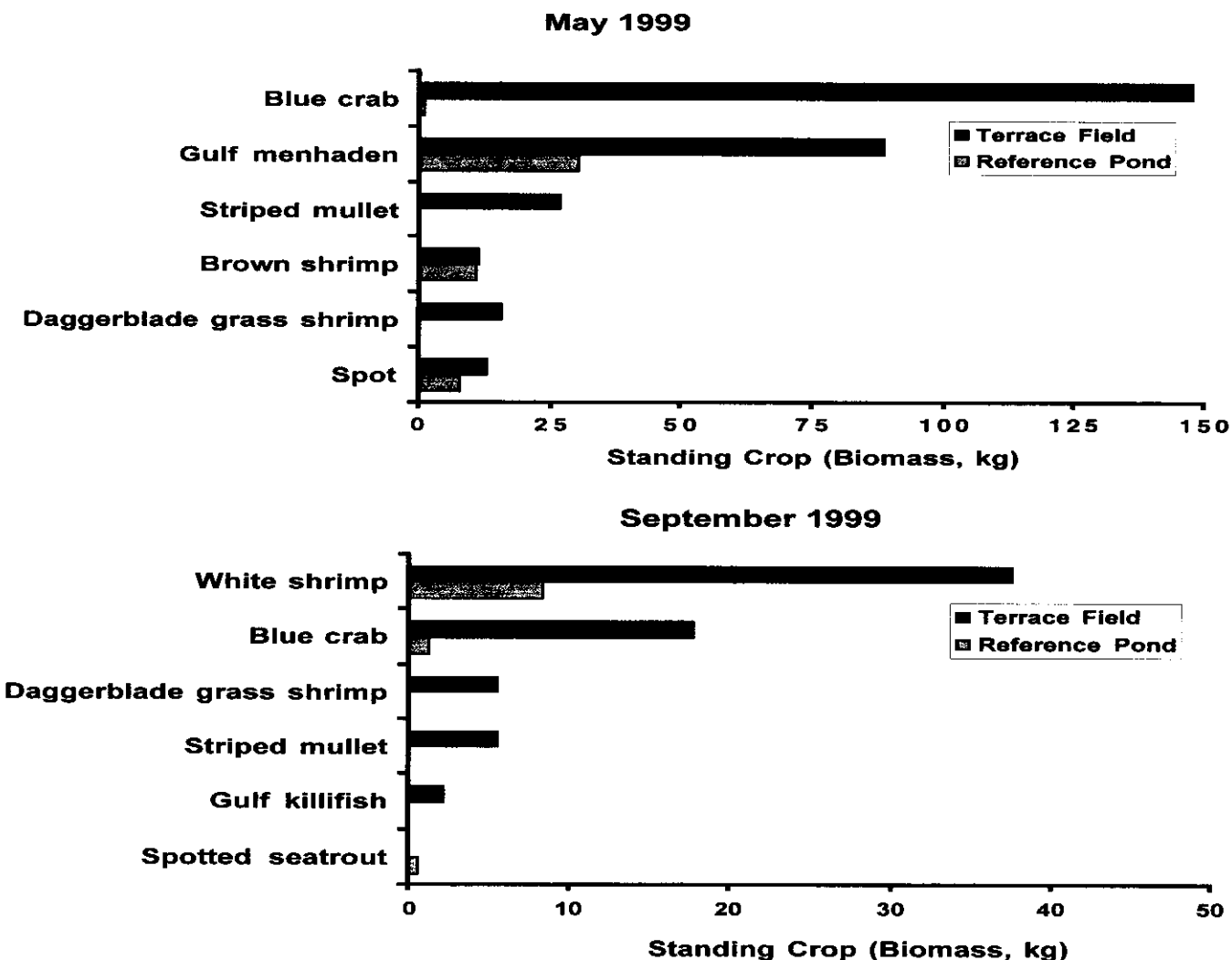


Figure 4. Comparison of hypothetical 1-ha terrace field and reference pond in May and September 1999 using estimated standing crop (total biomass of organisms, kg) of selected species. Animal biomasses used in the calculations are means determined from sample collections in each habitat type (terrace marsh and terrace pond, reference pond). Design of the terrace field in this comparison is based on the marsh terraces in our study area where marsh and ponds occupy 21% and 79% of the total area, respectively.

supported much greater biomasses than the reference pond for most species. Exceptions were brown shrimp (mean biomass in terrace field and reference pond was similar in May) and spotted seatrout (not collected in either terrace habitat type in September).

Our results indicate that redesigning terrace fields to increase the proportion of emergent marsh can enhance standing crops of species that select for marsh vegetation. Our estimate of brown shrimp standing crop in May was slightly lower in 1 ha of Sabine terrace field than in a marsh pond of equal size (Figures 3 and 5). When we simulated the effect of changing the area of marsh in a terrace field, we found that increasing the marsh area from 21% to 26% increased the standing crop of brown shrimp to the level of the marsh pond (Figure 5).

Few of the species we tested differed in size among treatments. Within marsh treatments for September, white shrimp were significantly larger in the reference marsh, whereas blue crab were larger in the terrace marsh (Table 4). Most species were taken too infrequently in terrace ponds and the reference pond to include these treatments in the size analysis. However, brown shrimp were abundant enough in the reference pond in May, and white shrimp and blue crab were collected frequently enough in both terrace ponds and the reference pond in September for analysis. In May, brown shrimp did not differ significantly in size among treatment areas (ANOVA, d.f.=2,30,  $F=1.677$ ,  $p=0.2039$ ). In September, the mean size (20.1 mm) of white shrimp at reference pond sites was not significantly different from those at either terrace marsh

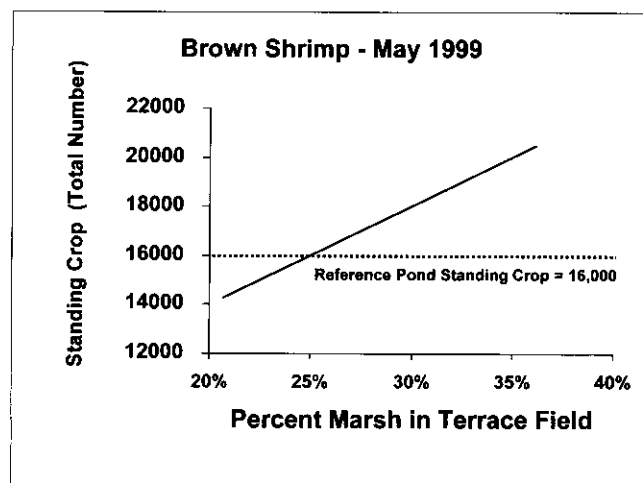


Figure 5. An illustration of how increasing the proportion of marsh in a terrace field affects standing crop (total numbers) of brown shrimp in spring. Animal densities used in the calculations are mean brown shrimp densities determined from May 1999 samples collected in each habitat type (terrace marsh, terrace pond, and reference pond). When 21% of the terrace field is marsh (as in our study area), brown shrimp standing crop is less in the terrace field than in a marsh pond of equal size. Standing crop in the terrace field equals that in a marsh pond when 26% of the terrace field is marsh and exceeds the pond standing crop when > 26% of the terrace field is marsh.

(ANOVA contrast, d.f.=1,55,  $F=0.921$ ,  $p=0.3413$ ) or terrace pond (mean=21.1 mm) sites (ANOVA contrast, d.f.=1,55,  $F=0.376$ ,  $p=0.5424$ ). Blue crab were significantly larger at terrace marsh than reference pond sites (means: 16.8 mm vs 8.7 mm, ANOVA contrast, d.f.=1,48,  $F=24.280$ ,  $p=0.0001$ ), although not significantly different between terrace pond (mean=10.4 mm) and reference pond sites (ANOVA contrast, d.f.=1,48,  $F=1.367$ ,  $p=0.2480$ ).

Submerged aquatic vegetation was not widely dis-

tributed in the study area, but *Ruppia maritima* was present in some terrace ponds and in the reference pond. Based on the occurrence of SAV at our sample sites, we estimated the portion of total bottom area occupied by *Ruppia* to be 0.1% (May) and 4.8% (September) in the terrace ponds and 0.2% (May) and 0.3% (September) in the reference pond. Even when most abundant in September, the distribution of *Ruppia* within the terrace ponds was very patchy. Because SAV was present in so few samples, we did not analyze these data further.

We found measurable differences in environmental conditions among some of the treatment areas (Table 5). Reference pond sites generally had higher dissolved oxygen concentrations, deeper water, and were located farther from the shoreline than terrace pond or terrace marsh sites (Table 5). In contrast, terrace marsh and reference marsh sites had similar environmental characteristics; none of the means of the variables we measured (except salinity in May) were significantly different between the two marsh treatments (Table 5).

## DISCUSSION

The marsh terraces at Sabine NWR provided habitat support for some species that was equivalent to natural areas. In our study, terrace marsh supported densities of white shrimp similar to those of a natural marsh, and both white shrimp and gulf menhaden were more abundant within terrace ponds than in a nearby natural pond. Moreover, the abundance and biomass of species that use the marsh surface were generally much greater in terrace marsh than over non-vegetated pond bottom; a terrace field would support higher standing crops (numbers and biomass) of most of these species than a shallow marsh pond of similar size. When white shrimp, brown shrimp, and blue crab were abundant in the study area, their mean densities were 8.0, 2.9,

Table 4. Comparison of sizes (mean  $\pm$  1 SE) in mm of numerically dominant animals collected at marsh sites in May and September 1999. Each mean (total length of shrimps or carapace width of crabs) for a marsh treatment was estimated from the mean sizes of  $n$  samples that contained that species. Degrees of freedom (df),  $F$ - and  $P$ -values listed are from an ANOVA comparing mean size in the two marsh treatments. An \* indicates that the probability value was significant after alpha was adjusted as described by Rice (1989).

Species	Terrace Marsh			Reference Marsh			df	F	P
	Mean	SE	n	Mean	SE	n			
<i>May</i>									
Brown shrimp	35.9	(4.64)	10	46.1	(2.90)	10	1,18	3.476	0.0786
Blue crab	29.3	(8.79)	9	17.4	(1.04)	9	1,16	1.833	0.1946
Daggerblade grass shrimp	26.0	(1.53)	10	29.7	(1.11)	10	1,18	3.773	0.0679
<i>September</i>									
White shrimp	22.0	(1.75)	10	33.4	(2.14)	10	1,18	17.312	0.0006*
Blue crab	16.8	(1.70)	10	12.2	(0.42)	10	1,18	6.949	0.0168
Daggerblade grass shrimp	17.5	(0.98)	10	20.7	(2.05)	10	1,18	1.989	0.1755

Table 5. Environmental characteristics of treatment areas. Mean and (1 SE) are given for variables measured in marsh vegetation and over non-vegetated bottom at terrace and reference areas sampled in May and September 1999. Each mean is estimated from 10 marsh sites or 20 samples taken in terrace or reference ponds (except Oxygen in May: Terrace marsh = 6, Reference marsh = 4, Terrace pond = 9, and Reference pond = 9). Results (p values) are given for ANOVA analyses we used to compare means among the four treatment areas (except comparisons for stem density are between terrace marsh and reference marsh only) and *a priori* contrasts testing for significant differences between: 1 = terrace marsh and reference pond; 2 = terrace pond and reference pond; and 3 = terrace marsh and reference marsh. The ANOVA model used to do the analyses is shown in Table 1. An \* indicates probability value was significant for the overall ANOVA after alpha was adjusted as described by Rice (1989).

Variable	Terrace Marsh			Reference Marsh			Terrace Pond			Reference Pond			ANOVA p Value	Contrast p Values		
	Mean	SE		Mean	SE		Mean	SE		Mean	SE			(1) T Marsh vs R Pond	(2) T Pond vs R Pond	(3) T Marsh vs R Marsh
<i>May</i>																
SALINITY (‰)	16.8	(0.13)		15.4	(0.45)		16.8	(0.14)		15.8	(0.16)		0.0001*	0.0016	0.0003	0.0003
OXYGEN (ppm)	4.8	(0.64)		5.0	(0.52)		5.1	(0.37)		6.8	(0.04)		0.0019*	0.0011	0.0012	0.7851
WATER TEMPERATURE (°C)	24.4	(0.52)		25.5	(0.66)		24.8	(0.39)		24.6	(0.39)		0.5485			
WATER DEPTH (cm)	33.3	(2.17)		33.6	(2.35)		46.7	(1.00)		56.6	(2.38)		0.0001*	0.0001	0.0002	0.9322
TURBIDITY (FTU)	48.0	(11.25)		39.5	(14.99)		34.4	(6.96)		68.8	(12.27)		0.0906			
DISTANCE TO EDGE (m)	0.7	(0.06)		0.8	(0.08)		10.5	(1.50)		23.3	(2.99)		0.0001*	0.0001	0.0001	0.9817
STEM DENSITY (stems m <sup>-2</sup> )	74	(11.2)		66	(8.0)								0.5580			
<i>September</i>																
SALINITY (‰)	26.0	(0.37)		27.4	(0.43)		25.9	(0.24)		26.8	(0.35)		0.0139	0.1462	0.0346	0.0206
OXYGEN (ppm)	5.8	(0.56)		7.2	(0.38)		6.0	(0.45)		7.7	(0.20)		0.0029*	0.0036	0.0013	0.0590
WATER TEMPERATURE (°C)	30.6	(0.42)		31.2	(0.35)		31.5	(0.67)		32.2	(0.36)		0.2478			
WATER DEPTH (cm)	15.3	(2.15)		18.2	(1.53)		22.4	(1.09)		36.3	(1.04)		0.0001*	0.0001	0.0001	0.2143
TURBIDITY (FTU)	32.1	(12.87)		28.6	(13.09)		16.8	(2.35)		15.7	(1.90)		0.2232			
DISTANCE TO EDGE (m)	0.1	(0.10)		0.2	(0.11)		11.1	(1.11)		23.7	(3.19)		0.0001*	0.0001	0.0001	0.9798
STEM DENSITY (stems m <sup>-2</sup> )	110	(13.9)		101	(20.7)								0.7316			

and 2.4 times greater, respectively, in terrace marsh vegetation than in the reference pond. Because our study only examined one terrace field in comparison to one reference pond, it was not possible to examine variability among terrace fields and reference ponds. This problem with pseudoreplication (Hurlbert 1984) is not uncommon in assessments of created wetlands (Minello and Zimmerman 1992, Streever 2000). However, as documented in numerous studies elsewhere (Zimmerman and Minello 1984, Minello et al. 1994, Cicchetti 1998, Howe et al. 1999, Minello 1999, Rozas and Zimmerman 2000), penaeid shrimps and blue crab are strongly attracted to shoreline vegetation. Most of these species should clearly benefit by replacing some shallow, non-vegetated bottom with marsh terraces in estuaries where the area of emergent marsh has decreased and restoration by marsh terracing is practical.

Marsh terraces constructed in non-vegetated areas should provide animals with protection from large predators by adding vegetation structure in the form of emergent marsh on terrace ridges and, perhaps, by facilitating the growth of SAV within terrace ponds. Both emergent vegetation and submerged aquatics have been shown to reduce predation rates on species collected in our study (Heck and Thoman 1981, Minello and Zimmerman 1983, Wilson et al. 1987, 1990, Rozas and Odum 1988, Minello et al. 1989, Minello 1993). Although SAV was not widely distributed within the terrace ponds when we sampled the sites at Sabine NWR, *Ruppia* may be more widespread there at other times.

Local sediment processes also may be affected by the presence of terrace fields. Initial data collected at Sabine NWR indicated that turbidity was reduced within the terrace ponds compared with surrounding open-water areas (Underwood et al. 1991); results of extended monitoring, however, did not show a significant reduction in turbidity levels within the terrace ponds (LDNR 1993). In our study, the mean turbidity level in terrace ponds was about half that of the reference pond in May, but turbidity levels were not significantly different among treatments. In September, turbidity levels in ponds and the reference area were low and very similar. Some evidence for accretion within terrace ponds was reported initially, but subsequent and more-detailed analyses were inconclusive because most of the feldspar plots placed in the area to measure sediment accretion could not be located (LDNR 1993). Accretion rates do not appear to have been high enough to facilitate expansion of *Spartina* across the terrace ponds, and we saw no evidence that emergent vegetation will eventually cover the ponds. Initial monitoring of the Sabine NWR terracing did show that the terraces reduced fetch, wave height, and

shoreline retreat within the terraced marsh ponds (LDNR 1993).

The *Spartina* marsh established on terrace levees at Sabine NWR was not functionally equivalent to an adjacent natural marsh, even after 9 years of development. In our study, the terrace marsh supported brown shrimp, blue crab, daggerblade grass shrimp (in September), and brackish grass shrimp at densities about half or less those of the nearby natural marsh. Moreover, more striped mullet (May), white shrimp (September), and spotted seatrout (September) biomass was present in the natural marsh than in the terrace marsh. Similarly, densities of daggerblade grass shrimp and juvenile brown shrimp were significantly less in marshes constructed on dredged material than in natural marshes in Texas (Minello and Zimmerman 1992). In a more extensive study of 10 created marshes and five natural marshes in Galveston Bay, Minello and Webb (1997) reported that densities of brown shrimp, blue crab, and white shrimp in created marshes were only 25–40%, 5–56%, and 59%, respectively, of those in natural marshes. Interestingly, when they repeated the comparison for white shrimp after removing data from created marshes with relatively high surface elevations and low flooding duration, densities between created marshes and natural marshes were not significantly different (Minello and Webb 1997). This change did not occur for brown shrimp and blue crab; densities of these two species were still significantly greater in natural marshes even when the high-elevation created marshes were removed from the analysis. Perhaps white shrimp will use created marsh as readily as natural marsh when the marsh is constructed at an elevation similar to natural marsh. Results from other studies suggest that marsh surface elevation can influence habitat use by white shrimp and other species (Rozas and Reed 1993, Rozas and Zimmerman 2000). In our study, terrace marsh and reference marsh sites were similar in mean water depth, and mean densities of white shrimp in these two habitat types were nearly equal. Measures of water depth in our study are a good proxy for relative elevation because tidal fluctuations were low and our samples of the two marsh types were interspersed in time.

Comparisons of animal size between natural and created marshes may indicate differences in growth rates and habitat quality. Minello and Webb (1997) suggested that grass shrimp may have been smaller in created marshes due to inadequate food supply in these marshes. In our study, we detected few differences in the mean size of animals between the terrace marsh and the reference marsh. White shrimp, however, were significantly smaller in terrace marsh than natural marsh.

The terrace marsh was about 9 years old when we

collected our samples, but there is some evidence that age of created marshes has little relationship with habitat value for nekton. Recent studies in Galveston Bay, Texas showed that marsh established on dredged material develops rapidly and apparently reaches its maximum habitat support function for nekton in just over a year following construction (Goldberg 1996, Minello 2000). For some species, however, this level of habitat support may never reach equivalency with natural marshes (Minello and Webb 1997).

Design changes to increase the relative area of marsh in a terrace field should be able to enhance standing crops of species that select for marsh habitat. As an example, we simulated the effect of increasing marsh area on the standing crop of brown shrimp in a terrace field. Increasing the area of marsh within the terrace field from 21 to 26% would boost the standing crop of brown shrimp in spring, making it equivalent to the marsh pond standing crop. Increasing the percentage of marsh in the terrace field to 35% would increase the standing crop to 20,000 brown shrimp  $\text{ha}^{-1}$ , a 25% increase over the level in a marsh pond. The area of marsh could be enlarged by increasing the density of terrace ridges and reducing the size of cells in a terrace field. However, other design changes also are possible. For example, terrace ridges can be laid out in various patterns other than the checkerboard form. Design considerations for maximizing the habitat function of marsh terraces should be assessed in future research.

Interestingly, the white shrimp densities we observed in our study are some of the highest reported in coastal environments. White shrimp occurred at mean densities of  $> 60 \text{ m}^{-2}$  in both of our marsh treatments, and four of our samples within *Spartina* vegetation contained  $> 100$  individuals  $\text{m}^{-2}$ . Other studies of shoreline *Spartina* marsh using similar gear and methods have reported mean white shrimp densities as high as  $49.6 \text{ m}^{-2}$  (Rozas and Minello 2000) and  $30.0 \text{ m}^{-2}$  (Minello and Webb 1997) in Galveston Bay, Texas. Lower densities were reported for Mobile Bay ( $1.4 \text{ m}^{-2}$ , Howe et al. 1999) and other locations in Texas and Louisiana ( $1.6$ – $26.7 \text{ m}^{-2}$ , Zimmerman et al. 2000). Densities of other decapod crustaceans (brown shrimp, pink shrimp, blue crab, grass shrimp) collected in our study were similar to those found in other studies of similar habitat.

In summary, marsh terracing appears to reduce fetch, wave energy, and shoreline erosion within coastal water bodies. Terracing also may provide favorable conditions within ponds for SAV establishment and growth, but we found no evidence for increased SAV cover in the terrace fields at Sabine NWR. For many nekton species, marsh terracing appears to increase standing crop measured in both numbers and biomass

per hectare. Densities of white shrimp, brown shrimp (spring), and blue crab were much greater in terrace marsh than over non-vegetated mud bottom, the habitat type that is replaced when marsh-terracing projects are constructed. The terrace marsh at Sabine NWR, however, was not functionally equivalent to nearby natural marsh based on densities of brown shrimp, blue crab, and grass shrimps or biomass of white shrimp, striped mullet, and spotted seatrout. Future terracing plans could enhance the habitat value of projects for fishery species by increasing the proportion of marsh in terrace fields. We believe that marsh terracing can be used successfully as a restoration technique to enhance coastal fisheries.

## ACKNOWLEDGMENTS

This research was conducted through the Fishery Ecology Branch (FEB) of the Galveston Laboratory in the Southeast Fisheries Science Center (NOAA, National Marine Fisheries Service). The assistance of everyone in the FEB was essential for the successful completion of this project. In particular, we thank Heather Carter, Jim Ditty, Jennifer Doerr, David Evans, Kirk Kilfoyle, Seth King, and Shawn Hillen for helping collect and process samples and Phillip Caldwell for producing Figure 2. We also thank Parcy Abohweyere (Nigerian Institute of Oceanography and Marine Research), Cherie O'Brien (Texas Parks and Wildlife Department), and Bill Streever (U.S. Army Corps of Engineer Waterways Experiment Station), who volunteered to help us collect the nekton samples. We gratefully acknowledge the logistical support and information about the Sabine National Wildlife Refuge provided by Chris Pease, Herb Bell, Ben Mense, Roy Walker, and other refuge personnel. Thanks to Ron Boustany and Tommy Michot (USGS, National Wetlands Research Center) for providing the photograph for Figure 1. The suggestions of Bill Good, William J. Richards, Greg D. Steyer, Bill Streever, and two anonymous reviewers improved this paper. We acknowledge the U.S. Army Corps of Engineer Waterways Experiment Station and the National Marine Fisheries Service for funding this project.

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